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# Macroinvertebrate Communities in Agriculturally Impacted Southern Illinois Streams: Patterns with Riparian Vegetation, Water Quality, and In-Stream Habitat Quality

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## ABSTRACT

**Relationships between riparian land cover, in-stream habitat, water chemistry, and macroinvertebrates were examined in headwater streams draining an agricultural region of Illinois. Macroinvertebrates and organic matter were collected monthly for one year from three intensively monitored streams with a gradient of riparian forest cover (6, 22, and 31% of riparian area). Bioassessments and physical habitat analyses were also performed in these three streams and 12 other nearby headwater streams. The intensively monitored site with the least riparian forest cover had significantly greater percent silt substrates than the sites with medium and high forest cover, and significantly higher very fine organics in substrates than the medium and high forested sites. Macroinvertebrates were abundant in all streams, but communities reflected degraded conditions; noninsect groups, mostly oligochaetes and copepods, dominated density and oligochaetes and mollusks, mostly *Sphaerium* and *Physella*, dominated biomass. Of insects, dipterans, mostly Chironomidae, dominated density and dipterans and coleopterans were important contributors to biomass. Collector–gatherers dominated functional structure in all three intensively monitored sites, indicating that functional structure metrics may not be appropriate for assessing these systems. The intensively monitored site with lowest riparian forest cover had significantly greater macroinvertebrate density and biomass, but lowest insect density and biomass. Density and biomass of active collector–filterers (mostly *Sphaerium*) decreased with increasing riparian forest. Hilsenhoff scores from all 15 sites were significantly correlated with in-stream habitat scores, percent riparian forest, and orthophosphate concentrations, and multiple regression indicated that in-stream habitat was the primary factor influencing biotic integrity. Our results show that these "drainage ditches" harbor abundant macroinvertebrates that are typical of degraded conditions, but that they can reflect gradients of conditions in and around these streams.**

Due to limitations associated with standard chemical monitoring programs and the inherent benefits of biological assessments, the use of freshwater organisms as indicators of environmental quality has recently increased (e.g., Abel, 1989; Rosenberg and Resh, 1993; Loeb and Spacie, 1994; Barbour et al., 1999). A wealth of information regarding tolerances of macroinvertebrate taxa to various disturbances and pollutions exist (e.g., Hynes, 1960; Chutter, 1972; Hilsenhoff, 1987, 1988; Winner et al., 1980; Barbour et al., 1999), and this has given rise to macroinvertebrate-based bioassessment methods that are sensitive to a variety of disturbances and may be applicable in many situations and regions (e.g., Lenat, 1988; Lang et al., 1989; Plafkin et al., 1989; Kerans and Karr, 1994; Barbour et al., 1999). However, bioassessment of freshwater habitats in a particular region often requires development of suitable methods and metrics based on knowledge of communities in specific systems (e.g., Barton, 1996).

Agricultural runoff is a major contributor to degradation of aquatic ecosystems in the United States and has deleterious effects on stream water quality and in-stream habitats (USEPA, 1994). The effects of agricultural practices on streams include changes in riparian vegetation, alteration of channel morphology, degraded in-stream habitats, and higher sediment and nutrient loads relative to unimpacted systems (Cooper, 1993). These impacts are apparent in streams that drain agriculturally dominated landscapes of the U.S. Midwest, including Illinois, where approximately 80% of the land surface is farmed (Illinois Department of Natural Resources, 1994).

In the past three decades, many studies have examined ways to minimize amounts of nutrients lost in agricultural runoff. Establishment of vegetated riparian buffers of grasses, trees, and shrubs adjacent to water bodies has become a widely accepted practice to reduce nutrient and sediment runoff into streams (Peterjohn and Correll, 1984; Lowrance et al., 1985; Jordan et al., 1993; Schultz et al., 1995; Dosskey, 2001). Research has shown that riparian buffer strips can significantly reduce nutrients and sediment in overland flow, improve in-stream habitat, and in turn improve biotic integrity and thus ecosystem health (Todd et al., 1983; Dillaha et al., 1989; Osborne and Kovacic, 1993; Sweeney, 1993; Davies and Nelson, 1994; Vought et al., 1994; Naiman and Decamps, 1997; Lee et al., 1999; Weigel et al., 2000; Whiles et al., 2000). However, few studies have examined riparian buffer effectiveness for reduction of nutrient and sediment movements into streams at the watershed scale (e.g., Jones et al., 2001), and the intensively agricultural regions of Illinois are no exception (Illinois Department of Natural Resources, 1999).

Given the general lack of information on aquatic communities found in the highly degraded streams that typify the agricultural U.S. Midwest, and the need for quantitative information on relationships between riparian land use, in-stream habitat quality, water chemistry, and biotic integrity, our objectives were to (i) characterize and quantify aquatic macroinvertebrate communities in streams of southern Illinois' agriculturally dominated landscape; and (ii) identify riparian and in-stream fac-

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**Abbreviations:** AFDM, ash-free dry mass; CPOM, coarse particulate organic material; EPT, Ephemeroptera, Plecoptera, and Trichoptera; FPOM, fine particulate organic material; VFPOM, very fine particulate organic material.

tors influencing macroinvertebrate communities, and thus biotic integrity. We predicted that macroinvertebrate communities in these systems would be characteristic of highly degraded habitats (e.g., low diversity and dominated by disturbance-tolerant taxa), but that they would reflect gradients of conditions found in and along differentially impacted streams.

## MATERIALS AND METHODS

### Study Area

The Sugar Creek watershed is located in the southern Illinois counties of Clinton, Bond, and Madison (Fig. 1), approximately 25 km east of St. Louis, Missouri. The watershed is not tile-drained and has a variety of silt-loam soil associations. The primary land use within the watershed is agriculture, including row crops and some dairy farming. Major apparent disturbances to streams in the watershed include channelization, sedimentation, and loss of riparian vegetation. Riparian forests, where present, are composed of a variety of early successional mixed hardwood species including black willow (*Salix nigra* Marsh.), box elder (*Acer negundo* L.), and common hackberry (*Celtis occidentalis* L.). Annual precipitation during the 2001 study year was about 40 mm below the long-term (1950–2001) average of 970 mm/yr.

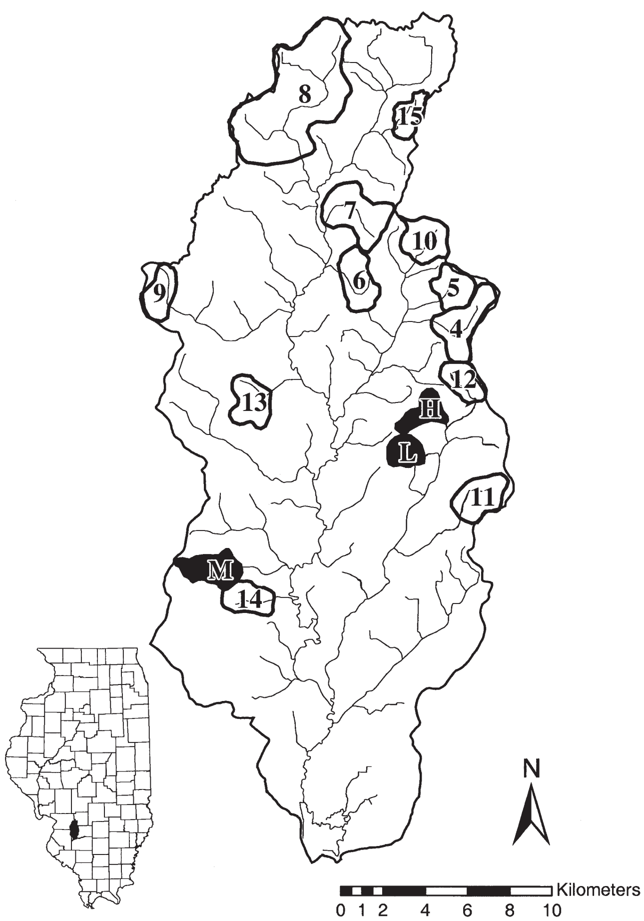


**Fig. 1. Map of all study streams, subwatersheds, and the three intensively monitored watersheds within the Sugar Creek watershed, Clinton, Bond, and Madison counties, Illinois. Letters L, M, and H indicate the intensively monitored streams with low riparian forest cover (6%), medium riparian forest cover (22%), and high riparian forest cover (31%), respectively, and these watershed areas are indicated in black. Numbers 4 to 15 indicate the streams included for the rapid bioassessment, and these watershed areas are outlined in black.**

Fifteen headwater streams and subwatersheds in the Sugar Creek watershed were selected for water quality monitoring and bioassessment using ArcView GIS 3.2 (ESRI, 1999) (Fig. 1; Appendix). These streams drain similar-sized watersheds dominated by row crop agriculture and are located in Clinton and Madison counties. Digital orthophoto quadrangles with a 1-m ground resolution (quarter-quadrangle image cast on the Universal Transverse Mercator Projection on the North American Datum of 1983 [USGS, 2000]) and stream network maps (county digital stream cartographic layers provided by the Illinois Natural Resources Geospatial Data Clearinghouse derived from the United States Geological Survey 1:100 000 Digital Line Graph file hydrography layer 1980–1986 [Illinois Department of Natural Resources, 1994]) were used to identify perennial stream channels. All visible drainage networks, including first- and second-order intermittent streams within the subwatersheds, were digitized.

A 15-m buffer zone was delineated on each side of a digitized 100-m stream segment and the total land surface area within the 15-m buffer was estimated. There were no row crops present in the delineated buffers and all other vegetation consisted of exotic cool-season grasses and a variety of forbs and shrubs. The presence of forest canopy within the 15-m riparian buffer zone was also digitized from digital orthophoto quadrangles to derive the forested area within the buffer zone. Total forested area within the 15-m buffer zone was divided by the total area of the buffer zone to calculate the proportion of forest area within a 15-m riparian buffer zone (Appendix). Buffers of 15 m were chosen based on field-scale riparian studies that showed significant (>50%) nutrient and sediment attenuation in the first 15 m of buffer zones (e.g., Dillaha et al., 1989; Dosskey, 2001).

Three of the 15 watersheds were selected for intensive monitoring and are indicated in bold (Fig. 1). These three intensively monitored sites have similar land use (predominately row crop agriculture) and catchment areas. They also represent a gradient of riparian forest cover in their respective buffers ranging from low (6%), to medium (22%), to high (31%), and are referred to herein as low, medium, and high, respectively (Table 1).

**Table 1. Physical characteristics of the three intensively monitored streams in the Sugar Creek watershed, Clinton and Madison counties, Illinois.**

| | Cover† | | |
|---|---|---|---|
| **Parameter** | **Low** | **Medium** | **High** |
| **Catchment area, ha** | **220** | **369** | **250** |
| **Row crop in watershed, %** | **90** | **89** | **82** |
| **Stream length, m** | **4707** | **12 300** | **5208** |
| **Forest in 15-m buffer, %** | **6** | **22** | **31** |
| **Average wetted width, m** | **2** | **4** | **3** |
| **Substrate composition, %** | | | |
| **Gravel** | **17** | **58** | **35** |
| **Sand** | **28** | **26** | **51** |
| **Silt** | **55a‡** | **16b** | **14b** |
| **Temperature** | | | |
| **Average, °C** | **10.9** | **10.9** | **8.3** |
| **Minimum, °C** | **0.0** | **0.0** | **0.3** |
| **Maximum, °C** | **28.7** | **26.1** | **27.0** |
| **Degree days** | **2442** | **3121** | **1346** |
| **Flow, $m^3/s$** | | | |
| **Average** | **0.1** | **0.4** | **0.2** |
| **Maximum** | **0.8** | **2.3** | **1.9** |
| **Minimum** | **0.0** | **0.0** | **0.0** |

**† Low, 6% riparian forest; medium, 22% riparian forest; high, 31% riparian forest cover along streams.**

**‡ Different letters within rows indicate significant differences among low, medium, and high sites (paired *t* test, $\alpha = 0.05$).**

## Intensively Monitored Streams

### Macroinvertebrates

Sampling reaches for the three intensively monitored streams were 100 m and these streams and sampling reaches were used in the rapid bioassessment. Aquatic macroinvertebrates were collected monthly from January 2001 to January 2002 when stream reaches contained water. On each sampling date, four samples were collected with a 20-cm-diameter stovepipe coring device from randomly selected locations in the channel along each 100-m reach. The corer was pushed into the substrates to a depth of at least 10 cm and all water and the upper 10 cm of substrate within the corer was removed with a cup and 250-μm mesh hand net and placed into a 19-L bucket. Material in the bucket was then elutriated through a 250-μm sieve until visual inspection indicated that only inorganic materials remained in the bucket. Material retained on the sieve was placed in a plastic bag and preserved in 8% formalin containing Phloxine-b dye to aid in sorting.

In the laboratory, samples were rinsed through nested 4-mm, 1-mm, and 250-μm sieves. The material retained on the 4- and 1-mm sieves, coarse particulate organic material (CPOM, >1 mm), was examined under a dissecting microscope and all macroinvertebrates were removed and placed in labeled vials containing 8% formalin. Fine particulate organic material (FPOM, <1 mm > 250 μm) that was retained on the 250-μm filter was occasionally subsampled (usually 1/2–1/16) using a Folsom plankton wheel before removing macroinvertebrates under a dissecting microscope. All macroinvertebrates were identified to the lowest practical taxonomic level; insects were usually identified to genus and noninsects to order. Chironomidae were classified as either predatory (Tanypodinae) or nonpredatory.

All invertebrates were measured (total body length) to the nearest millimeter. Biomass (mg) was estimated using taxon-specific length–mass relationships obtained from Benke et al. (1999) or regressions that we made for regional taxa using the same procedures as Benke et al. (1999). Functional feeding group designations were based on Merritt and Cummins (1996) or regional studies of local taxa. Shannon diversity ($H'$, calculated with log base 10), percent dominant taxon, and taxa richness were calculated according to Brower et al. (1997). The EPT index was calculated as the number of Ephemeroptera, Plecoptera, and Trichoptera taxa present. Tolerance values for a modified Hilsenhoff index were from a comprehensive study by the Nebraska Department of Environmental Quality (1991). We used Nebraska Department of Environmental Quality (1991) values because they were derived from similar, primarily agricultural, streams that are subjected to a similar array of disturbances and have similar taxonomic composition to ours. Tolerance values and Hilsenhoff index scores range from 0 (lowest tolerance to pollution = best possible conditions) to 5 (highest tolerance = highly degraded conditions).

Substrate composition was estimated in each benthic core sample after elutriation and removal of macroinvertebrates and organic material. The remaining mineral material in the bottom of the bucket was visually examined and assigned a percent particle size composition based on a modified Wentworth scale (Cummins, 1962). Temperature data loggers recorded water temperature at 2-h intervals in each of the three intensively monitored streams throughout the study.

### Benthic Organic Material

Benthic organic matter was estimated in each of the three intensively monitored streams seasonally (February, May, August, and November 2001) using the same stovepipe core samples and sieving techniques as for macroinvertebrates. Organic material retained on the 4-mm sieve was sorted into recognizable categories of roots, wood, grass, seeds and fruits, leaves, and corn (stalks, kernels, and cobs); anything unrecognizable was classified as miscellaneous CPOM. Material retained on the 1-mm sieve was scanned for recognizable material to be placed in the above categories and the rest was added to the miscellaneous CPOM category. All material retained on the 250-μm sieve was considered miscellaneous FPOM. Categories of CPOM and FPOM were placed into aluminum weighing dishes and dried at 50°C in a drying oven for 48 h and weighed. Samples were cooled in a desiccator, weighed to the nearest thousandth of a gram, and then ashed in a 500°C muffle furnace for approximately one hour. Samples were returned to the drying oven for 48 h and reweighed to estimate grams ash-free dry mass (AFDM) and corrected for area sampled to yield g AFDM/m$^2$.

For very fine particulate organic material (VFPOM, <250 μm > 1.6 μm), a subsample of material that passed through the 250-μm sieve was collected in the field. For this procedure, the volume of water and associated materials removed from the core and placed in the bucket was recorded. The sample was then agitated to suspend all materials and poured through a 250-μm sieve into a sample bottle to obtain an approximately 500-mL subsample. In the laboratory, subsamples were resuspended and 10 to 50 mL of the slurry was vacuum-filtered through pre-ashed and weighed glass fiber filters. Filters were then placed in a 50°C drying oven for approximately 48 h, cooled in a desiccator, weighed to the nearest 0.1 mg on an analytical balance, and ashed in a 500°C muffle furnace for approximately 1 h. Filters were then rewetted with distilled water and returned to the drying oven for 48 h, then reweighed to estimate AFDM. Values were corrected for original volumes of material in cores and area sampled by the core to yield g AFDM/m$^2$.

### Data Analysis

Statistical procedures were performed using JMP 4.02 (SAS Institute, 2000). Paired differences of monthly means from the three intensively monitored streams were compared using a paired differences procedure based on Stewart-Oaten et al. (1986). For this procedure, a $t$ test was used to test the null hypothesis that the differences between paired monthly means from two streams were not different from zero. This form of statistical analysis was used to reduce problems associated with temporal autocorrelation in ecological time-series data, because paired differences are likely to have less autocorrelation (Stewart-Oaten et al., 1986). Strictly speaking, statistical inferences based on this procedure are limited to differences between the three intensively sampled streams. To reduce odds of Type I errors, paired comparison tests were limited to major categories of organic matter and more abundant or dominant macroinvertebrates.

Paired differences between any two sites could only be examined when water was present in both. Comparisons between the low cover and the medium cover sites were thus based on samples from January, February, March, April, May, August, November, and December 2001, and January 2002. Paired differences between the low cover and the high cover sites included January, February, March, April, May, and December 2001, and January 2002, and comparisons between the medium cover and the high cover sites were based on January, February, March, April, May, and December 2001, and January 2002.

## Rapid Bioassessment

### Physical Habitat Analysis and Water Quality

Physical habitat scoring was performed in each of the 15 streams in June of 2001 following the USEPA's standard protocols for quantifying in-stream and streamside habitats (Barbour et al., 1999) on a 100-m study reach at the downstream end of each subwatershed. A composite score (ranging from 0 to 100, 0 indicating poorest physical habitat quality and 100 indicating optimal physical habitat quality) was calculated using the following parameters for low gradient streams: epifaunal substrate/available cover, pool substrate characterization, sediment deposition, channel alteration, channel sinuosity, bank stability, vegetative protection, and riparian vegetative zone width. At each stream, physical habitat scores estimated by two observers were averaged to reduce bias. In a companion study, monthly grab samples were collected from the 15 sites during baseflow conditions from May 2001–April 2002 and analyzed for dissolved nitrate N, ammonium N, and orthophosphate P (Webber et al., 2003).

### Macroinvertebrate Communities

Macroinvertebrate-based rapid bioassessments were conducted in 50-m study reaches of each of the 15 sites in May 2001 following the USEPA's multihabitat procedure for low gradient streams (Barbour et al., 1999). All study reaches began at least 50 m upstream from any road bridges, when present, and proceeded upstream. All but one of the 15 study streams had only pool and run habitats dominated by fine substrates. One stream had a small riffle area, which was sampled in proportion to availability. Samples were collected with a 500-μm mesh dip net that was used to collect a total of 20 jabs (one jab = approximately 0.5-m length movement of the net along the substrates), which were taken over the length of each reach, beginning at the downstream end. Samples were emptied into a 19-L bucket, elutriated through a 500-μm sieve, placed in a plastic bag, and preserved in 8% formalin containing Phloxine-b dye.

A 225-count random subsample of macroinvertebrates was removed from each sample using a gridded and numbered sorting pan and table of random numbers (Barbour et al., 1999). Macroinvertebrates were identified to the lowest taxon possible using a dissecting scope; generally, insects were identified to genus and noninsects were identified to order. Taxa richness and percent dominant taxon were calculated according to Brower et al. (1997), and a modified Hilsenhoff biotic index and EPT index were calculated as described above for the intensively monitored site samples. Percent insect, oligochaete, and active filterer densities were also calculated.

### Data Analysis

Simple linear correlation and multiple regression were used to examine relationships between riparian vegetation, water chemistry, in-stream physical habitat quality, and macroinvertebrate community parameters among the 15 study reaches. These analyses were performed using the JMP 4 statistical package (SAS Institute, 2000).

## RESULTS

## Intensively Monitored Streams

### Substrates and Organic Material

Silt substrates dominated the low cover site, whereas sand and gravel were more prevalent in the medium and high cover sites (Table 1). Percent silt substrates in the low cover site was significantly greater than the medium ($P = 0.028$) and high ($P = 0.047$) cover sites (Table 1).

Average benthic VFPOM ranged from 5499 g AFDM/m$^2$ in the high cover site to 14 452 g AFDM/m$^2$ in the low cover site, and the low cover site had significantly greater VFPOM than the medium ($P = 0.031$) and the high ($P = 0.043$) cover sites (Table 2). Average benthic FPOM values ranged from 91 g AFDM/m$^2$ in the medium cover site to 182 g AFDM/m$^2$ in the low cover site, and the low cover site had significantly greater FPOM ($P = 0.039$) than the medium cover site (Table 2). Mean total CPOM ranged from 306 g AFDM/m$^2$ in the medium cover site to 892 g AFDM/m$^2$ in the low cover site, and the low cover site had significantly more total CPOM than both the medium ($P = 0.028$) and high cover ($P = 0.042$) sites (Table 2). Within the CPOM category, the low cover site had more corn and grass material than the medium cover site, and the high cover site had more root material than both the low and the medium cover sites (Table 2). There were also trends of more miscellaneous CPOM, wood, and seeds and fruits in the low cover site compared with the medium and the high cover sites.

**Table 2. Average values for benthic organic materials (±1 standard error) in the three intensively monitored streams in the Sugar Creek watershed.**

| | Cover‡ | | |
|---|---|---|---|
| Category† | Low | Medium | High |
| | g ash-free dry mass (AFDM)/m$^2$ | | |
| VFPOM | 14 451.6 (6087.8)a§ | 9110.4 (3977.3)b | 5499.0 (2183.8)b |
| FPOM | 182.1 (24.3)a | 91.1 (30.8)b | 160.5 (31.5)a,b |
| Total CPOM | 891.7 (459.2)a | 305.8 (80.7)b | 327.0 (72.1)b |
| Miscellaneous CPOM | 626.3 (466.9) | 217.1 (72.4) | 167.3 (45.7) |
| Roots | 0.0 (<0.1) | 0.6 (0.6) | 38.8 (10.8) |
| Wood | 200.4 (106.8) | 43.4 (21.1) | 94.8 (28.3) |
| Grass | 13.8 (4.2) | 0.5 (0.5) | 1.3 (0.6) |
| Seeds, fruits | 15.8 (10.9) | 5.3 (4.2) | 5.2 (3.2) |
| Leaves | 8.9 (3.2) | 38.9 (34.5) | 8.9 (5.3) |
| Corn | 26.5 (9.2) | 0.0 (<0.1) | 10.7 (8.4) |

† VFPOM, very fine particulate organic material; FPOM, fine particulate organic material; CPOM, coarse particulate organic material.
‡ Low, 6% riparian forest; medium, 22% riparian forest; high, 31% riparian forest cover along streams.
§ Different letters within rows indicate significant differences among the three streams (paired *t* test, α = 0.05).

**Table 3. Annual average density (±1 standard error) of macroinvertebrate taxa for the three intensively monitored streams in the Sugar Creek watershed.**

| Taxon | Cover† Low | Medium | High |
|---|---|---|---|
| | no./m² | | |
| **Insects** | | | |
| Ephemeroptera | 107.4 (56.9)a‡ | 1.1 (1.2)a | 3.4 (1.9)a |
| Odonata | 16.0 (6.4) | 0.0 (<0.1) | 5.7 (2.9) |
| Trichoptera | 1.1 (1.2) | 1.1 (1.2) | 0.0 (<0.1) |
| Hemiptera | 1.1 (1.2) | 1.1 (1.2) | 0.0 (<0.1) |
| Megaloptera | 0.0 (<0.1) | 1.1 (1.2) | 0.0 (<0.1) |
| Coleoptera | 2.3 (1.6) | 3.4 (2.6) | 73.0 (40.6) |
| Diptera | 4620.5 (2016.4)a | 13 435.1 (4826.3)b | 12 565.1 (5456.9)a,b |
| Total | 4748.5 (2015.4)a | 13 443.1 (4825.5)b | 12 647.3 (5464.0)a,b |
| **Noninsects** | | | |
| Hydrazoa | 56.0 (41.0) | 166.9 (89.5) | 292.6 (262.0) |
| Turbellaria | 41.1 (27.2) | 0.0 (<0.1) | 2.3 (2.3) |
| Nematoda | 1355.5 (356.4)a | 426.3 (135.3)b | 1214.9 (472.3)a,b |
| Oligochaeta | 100 607.7 (30 411.3)a | 12 148.6 (4320.4)b | 13 341.0 (3755.4)b |
| Hirudinea | 18.3 (11.5) | 2.3 (1.6) | 11.4 (4.8) |
| Gastropoda | 297.2 (225.0)a | 9.1 (5.3)a | 37.7 (16.7)a |
| Bivalvia | 2971.1 (641.2)a | 81.1 (41.1)b | 27.4 (15.4)b |
| Ostracoda | 1526.9 (437.1)a | 52.6 (28.6)b | 349.7 (145.4)b |
| Copepoda | 15 760.0 (5092.2)a | 7505.1 (2434.9)b | 15 854.8 (7554.1)a |
| Amphipoda | 1.1 (1.2) | 1.1 (1.2) | 38.9 (37.2) |
| Isopoda | 8.0 (4.6) | 2.3 (2.3) | 0.0 (<0.1) |
| Decapoda | 3.4 (2.6) | 1.1 (1.2) | 0.0 (<0.1) |
| Total | 12 2646.4 (33 348.7)a | 20 396.6 (6182.7)b | 31 170.7 (10 867.5)b |

† Low, 6% riparian forest; medium, 22% riparian forest; high, 31% riparian forest cover along streams.
‡ Different letters within rows indicate significant differences among low, medium, and high sites (paired *t* test, $\alpha = 0.05$).

## Macroinvertebrates

Average total macroinvertebrate density ranged from 33 840 individuals/m² in the medium cover site to 127 395 individuals/m² in the low cover site, and was significantly greater in the low cover site compared with the medium ($P = 0.004$) and high ($P = 0.002$) cover sites. Average total biomass ranged from 1405 g AFDM/m² in the high cover site to 5041 g AFDM/m² in the low cover site, and was significantly greater in the low cover site compared with the medium ($P < 0.001$) and high ($P < 0.001$) cover sites. Despite higher total macroinvertebrate density and biomass, the low cover site had the lowest insect density and biomass of all three sites, and this was significantly lower ($P = 0.028$ and $P = 0.008$ for insect density and biomass, respectively) than the medium cover site (Tables 3 and 4).

Dipterans, mostly Chironomidae and Ceratopogonidae, dominated insect density and biomass in all three intensively monitored streams, and average dipteran density ($P = 0.028$) and biomass ($P = 0.005$) were both significantly greater in the medium cover site than in the low cover site (Tables 3 and 4). In contrast, density and biomass of Ephemeroptera (mostly Caenidae and Baetidae) and Odonata (mostly *Libellula*, *Enallagma*, and *Ischnura*) showed trends of higher values in the low cover site. Density and biomass of coleopterans, mostly Hydrophilidae and Dytiscidae, were generally greater in the high cover site (Tables 3 and 4). Density and biomass of Trichoptera, Hemiptera, and Megaloptera were very low in all sites.

Noninsects dominated macroinvertebrate density in all three streams, and average total noninsect density was significantly greater in the low cover site than the medium ($P = 0.003$) and the high ($P = 0.001$) cover sites (Tables 3 and 4). Noninsect biomass was also significantly greater in the low cover site than in the medium ($P < 0.001$) and the high ($P < 0.001$) cover sites. Oligochaetes and copepods dominated noninsect density, and oligochaetes and mollusks dominated biomass.

Average oligochaete density and biomass were signif-

**Table 4. Annual average biomass (±1 standard error) of macroinvertebrate taxa in the three intensively monitored streams in the Sugar Creek watershed.**

| Taxon | Cover† Low | Medium | High |
|---|---|---|---|
| | g ash-free dry mass (AFDM)/m² | | |
| **Insects** | | | |
| Ephemeroptera | 37.3 (23.4)a‡ | 0.5 (0.5)a | 5.1 (4.2)a |
| Odonata | 62.8 (33.1) | 0.0 (<0.1) | 6.6 (4.3) |
| Trichoptera | 0.6 (0.6) | 0.6 (0.6) | 0.0 (<0.1) |
| Hemiptera | 0.6 (0.6) | 0.6 (0.6) | 0.0 (<0.1) |
| Megaloptera | 0.0 (<0.1) | 2.0 (2.1) | 0.0 (<0.1) |
| Coleoptera | 6.3 (4.5) | 9.2 (8.4) | 168.7 (97.7) |
| Diptera | 267.3 (105.0)a | 912.1 (271.2)b | 446.8 (169.4)a,b |
| Total | 374.8 (115.4)a | 924.9 (269.8)b | 627.1 (263.9)a,b |
| **Noninsects** | | | |
| Hydrazoa | 0.1 (<0.1) | 0.2 (0.1) | 0.3 (0.3) |
| Turbellaria | 6.3 (3.4) | 0.0 (<0.1) | 0.4 (0.6) |
| Nematoda | 7.6 (2.0)a | 2.4 (0.8)b | 6.9 (2.7) a,b |
| Oligochaeta | 3144.5 (620.7)a | 597.4 (163.5)b | 514.8 (92.7)b |
| Hirudinea | 0.6 (0.5) | 0.2 (0.2) | 2.2 (2.1) |
| Gastropoda | 294.3 (87.6)a | 24.3 (15.0)b | 219.4 (120.3)a,b |
| Bivalvia | 1154.2 (280.0)a | 19.3 (7.9)b | 14.3 (9.2)b |
| Ostracoda | 8.9 (2.5)a | 0.3 (0.2)b | 2.0 (0.8)b |
| Copepoda | 15.8 (5.1)a | 7.5 (2.4)b | 15.9 (7.6)a |
| Amphipoda | 0.3 (0.3) | 0.9 (0.9) | 2.0 (1.4) |
| Isopoda | 20.5 (13.4) | 1.3 (1.3) | 0.0 (<0.1) |
| Decapoda | 12.7 (10.2) | 3.0 (3.0) | 0.0 (<0.1) |
| Total | 4665.7 (803.0)a | 656.7 (169.1)b | 778.2 (154.0)b |

† Low, 6% riparian forest; medium, 22% riparian forest; high, 31% riparian forest cover along streams.
‡ Different letters within rows indicate significant differences among low, medium, and high sites (paired *t* test, $\alpha = 0.05$).

**Table 5. Annual average density and biomass (±1 standard error) and percent contribution to total of each of macroinvertebrate functional feeding groups in the three intensively monitored streams in the Sugar Creek watershed.**

| Parameter | Cover† | | |
|---|---|---|---|
| | Low | Medium | High |
| | | **Density** | |
| Collector–gatherer, no./m$^2$ | 122 876.0 (3 434 647.0)a‡ | 31 936.0 (9207.4)b | 42 422.1 (15 722.9)b |
| Contribution to total, % | 96 | 94 | 97 |
| Collector–filterer, no./m$^2$ | 2975.7 (642.0)a | 238.9 (136.2)b | 91.4 (47.3)b |
| Contribution to total, % | 2 | <1 | <1 |
| Scraper, no./m$^2$ | 297.2 (225.0)a | 10.3 (6.3)a | 37.7 (16.7)a |
| Contribution to total, % | <1 | <1 | <1 |
| Shredder, no./m$^2$ | 0.0 (<0.1) | 0.0 (<0.1) | 2.3 (2.3) |
| Contribution to total, % | <1 | <1 | <1 |
| Predator, no./m$^2$ | 1297.5 (570.8) | 1675.4 (796.1) | 1264.4 (361.8) |
| Contribution to total, % | 1 | 5 | 3 |
| | | **Biomass** | |
| Collector–gatherer, mg AFDM§/m$^2$ | 34 226.6 (664.5)a | 1355.3 (284.9)b | 929.2 (214.3)b |
| Contribution to total, % | 95 | 87 | 66 |
| Collector–filterer, mg AFDM/m$^2$ | 1154.6 (279.6)a | 21.2 (8.5)b | 15.2 (9.5)b |
| Contribution to total, % | 3 | 1 | 1 |
| Scraper, mg AFDM/m$^2$ | 294.3 (87.6)a | 24.5 (15.1)b | 219.4 (120.3)a,b |
| Contribution to total, % | <1 | 2 | 16 |
| Shredder, mg AFDM/m$^2$ | 0.0 (<0.1) | 0.0 (<0.1) | 0.5 (0.5) |
| Contribution to total, % | <1 | <1 | <1 |
| Predator, mg AFDM/m$^2$ | 169.6 (65.1) | 159.8 (87.2) | 241.0 (87.2) |
| Contribution to total, % | <1 | 10 | 17 |

† Low, 6% riparian forest; medium, 22% riparian forest; high, 31% riparian forest cover along streams.
‡ Different letters within rows indicate significant differences among low, medium, and high sites (paired $t$ test, $\alpha = 0.05$).
§ Ash-free dry mass.

icantly greater in the low cover site than both the medium ($P = 0.005$ and $P = 0.001$, respectively) and high cover sites ($P < 0.001$ for both). Copepod density and biomass were both significantly reduced in the medium cover site compared with the low ($P = 0.058$ for both) and high ($P = 0.057$ for both) cover sites. Bivalves, mostly *Sphaerium*, were significantly more dense and had greater biomass in the low site than the medium ($P < 0.001$ for both) and high ($P < 0.001$ for both) cover sites. Gastropods, mostly *Physella*, were also generally more dense in the low cover site, and gastropod biomass was significantly higher in the low cover site than in the medium cover site ($P = 0.002$). This same trend of significantly greater values in the low cover site was evident in other, less dominant noninsect groups including nematodes and ostracods (Tables 3 and 4).

Collector–gatherers dominated functional structure at all three sites, with average percent contribution to density ranging from 94% in the medium cover site to 97% in the high cover site (Table 5). In contrast, shredders were virtually absent in all sites. Average collector–gatherer density and biomass were significantly greater in the low cover site than in the medium ($P = 0.004$ and $P = 0.002$, respectively) and high ($P = 0.002$ and $P < 0.001$, respectively) cover sites (Table 5). Filterer density and biomass were also significantly greater in the low cover site than the medium ($P < 0.001$ for both) and high ($P < 0.001$ for both) cover sites, and scraper biomass was significantly greater in the low cover site than in the medium cover site ($P = 0.002$). Predator density and biomass were fairly evenly distributed across sites.

Average modified Hilsenhoff index scores ranged from 3.9 in the low cover site to 3.6 in the medium and high cover sites, but were not significantly different across sites. However, the low cover site had significantly greater percent dominant taxon values than the medium ($P = 0.001$) and high ($P < 0.001$) cover sites, and significantly lower Shannon diversity than the medium ($P = 0.001$) and high ($P = 0.001$) cover sites (Table 6). Average taxa richness was low in all sites, and was significantly greater in the low cover site than the medium cover site ($P = 0.001$). The EPT values were extremely low and variable in all three sites, and showed a trend of higher values in the low cover site (Table 6).

## Rapid Bioassessment

Average percent dominant taxon (±1 standard error) was 61% (±5.0) and ranged from 23 to 93% across the 15 biological assessment sites. Taxa richness ranged from 4 to 12 and the mean for the 15 sites was 8.0 (±7.0). Average percent insect, percent Oligochaete, and percent active filterer densities were 52% (±8.0), 22% (±7.0), and 6% (±3.0) and ranged from 2 to 96, 0 to 79, and 0 to 38%, respectively. The EPT richness values were low, ranging from 0 to 2, and mean EPT richness was 0.2 (±0.1). Modified Hilsenhoff index scores ranged from 2.5 to 4.9 and averaged 3.5 (±0.2).

Percent dominant taxon and taxa richness showed no relationships with riparian vegetation, water chemistry, or physical habitat scores. Percent insect density showed nonsignificant trends of increasing with increasing physical habitat scores, increasing percent riparian forest cover, and decreasing orthophosphate concentrations. The EPT taxa richness values were too low and variable for meaningful statistical comparisons. Hilsenhoff index values, however, were significantly related to orthophosphate concentrations ($r^2 = 0.63$, $P = 0.0004$), percent riparian forest cover ($r^2 = 0.61$, $P = 0.0006$), and in-stream physical habitat scores ($r^2 = 0.72$, $P < 0.0001$)

**Table 6. Average values (±1 standard error) for macroinvertebrate community metrics in each of the three intensively monitored streams in the Sugar Creek watershed.**

| | Cover† | | |
|---|---|---|---|
| Metric | Low | Medium | High |
| Hilsenhoff index | 3.9 (0.1)a | 3.6 (<0.1)a | 3.6 (0.1)a |
| Dominant taxon, % | 87 (<0.1)a | 71 (<0.1)b | 65 (0.1)b |
| Shannon diversity ($H'$) | 0.4 (<0.1)a | 0.5 (<0.1)b | 0.5 (<0.1)b |
| Species richness | 10.7 (0.6)a | 8.7 (0.3)b | 9.3 (0.6)a,b |
| EPT taxa | 0.50 (0.1) | 0.07 (0.1) | 0.12 (0.1) |

† Low, 6% riparian forest; medium, 22% riparian forest; high, 31% riparian forest cover along streams.
‡ Different letters within rows indicate significant differences among low, medium, and high sites (paired $t$ test, $\alpha = 0.05$).

(Fig. 2). Multiple regression analysis with Hilsenhoff index scores and orthophosphate concentrations, percent riparian forest cover, and in-stream physical habitat scores (model: $R^2 = 0.8$, $P = 0.0004$) revealed that instream physical habitat was the most important variable ($P = 0.028$).

## DISCUSSION

### Intensively Monitored Streams

Our estimates of total macroinvertebrate density and biomass are in the range of, and even somewhat higher than, those from other similar-sized systems in the central United States. Using the same techniques and sieve sizes, Stagliano and Whiles (2002) estimated that annual average abundance and biomass of macroinvertebrates in a perennial reach of a tallgrass prairie stream on the Konza Prairie Biological Station (KPBS), Kansas, were approximately 24 000 individuals/m$^2$ and 2.30 g AFDM/m$^2$, respectively. Fritz and Dodds (2002) used similar techniques in a range of intermittent and perennial headwater sites on KPBS and also found generally lower densities than we observed. Using similar methods, but slightly larger mesh sizes, other lower macroinvertebrate densities were reported for an intermittent headwater stream in southern Oklahoma (Miller and Golladay, 1996) and a second-order stream in northern Kentucky (Johnson et al., 1994). Although macroinvertebrates were at least as abundant in our sites as other similar, less human-impacted headwaters, communities in the streams we examined were heavily dominated by pollution-tolerant taxa such as tubificid worms, fingernail clams, and pulmonate snails.

Although generally dominated by pollution-tolerant taxa, there were differences in macroinvertebrate communities across the gradient of conditions that we examined, indicating that communities in these systems can change and reflect ambient conditions. Macroinvertebrate density and biomass were higher in the low riparian forest site, but much of the abundance was tubificids, which are common inhabitants of very poor quality (e.g., low oxygen and dominated by silt) freshwater habitats (Hilsenhoff, 1987, 1988; Lenat, 1993; Brinkhurst and Gelder, 1991; Barbour et al., 1999). Chironomidae were the dominant insects in all sites, and percent insect contribution to density increased with increasing percent riparian forest cover, indicating that this metric may be

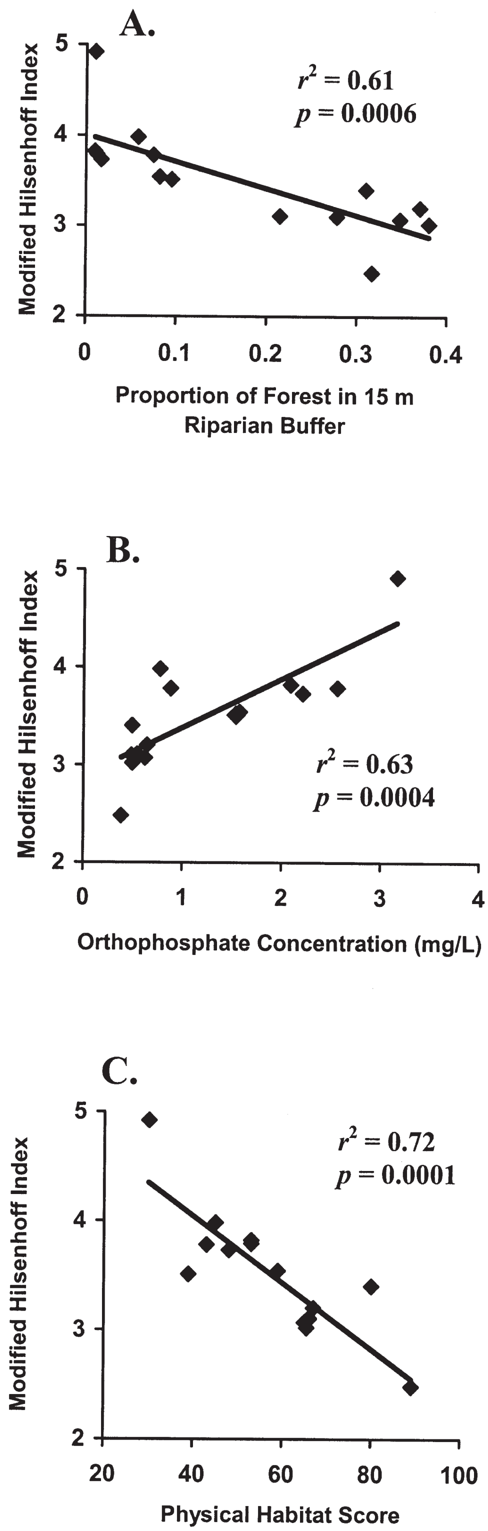


**Fig. 2. Linear regressions showing relationships between modified Hilsenhoff biotic index scores versus (A) proportion of forest cover in 15-m-wide riparian buffers along streams, (B) average orthophosphate concentration, and (C) in-stream physical habitat scores in the Sugar Creek watershed rapid bioassessment study sites.**

useful for bioassessment of degraded agricultural streams. In terms of functional structure, percent filterers also reflected the gradient in riparian forest cover. Filterers in these streams were dominated by fingernail clams (*Sphaerium*), which are active filter feeders commonly found in degraded areas with high amounts of fine sediment (Hilsenhoff, 1987, 1988; McMahon, 1991; Lenat, 1993; Barbour et al., 1999), and the contribution of *Sphaerium* declined precipitously from the low cover to high cover site.

Higher Ephemeroptera densities are typically associated with higher-quality conditions (Hilsenhoff, 1987, 1988; Lenat, 1993; Barbour et al., 1999). However, the mayflies we observed in these streams (*Baetis* spp., *Callibaetis* spp., and *Caenis* spp.) are relatively tolerant taxa that are commonly found in degraded streams (e.g., Hilsenhoff, 1987; Barbour et al., 1999). Thus, the nonsignificant trend of higher mayfly density in the low riparian cover site does not necessarily contradict other patterns observed during this study, and this trend also heavily influenced EPT metric patterns, a metric that proved unreliable (low values and high variability) in both the intensive examination of three streams and the bioassessment of the 15 streams.

It is widely accepted that the longitudinal nature of stream ecosystems results in physical habitat gradients, including the amount of riparian forest, which can sequentially alter biological communities and associated functional structure along the gradient. In particular, the river continuum concept (RCC; Vannote et al., 1980) predicts that shredders should be a dominant group in headwaters. These predicted patterns were not evident in these streams, as all of them, including the three intensively monitored sites and 12 other bioassessment sites, were dominated by collector–gatherers and shredders were poorly represented, even in sites with the highest riparian forest cover. This lack of conformity with predictions of the RCC is likely related to the high degree of human impacts on both physical (channelization, sedimentation) and chemical (nutrients, pesticides) habitat components. Further, this region was primarily tallgrass prairie, and even undisturbed systems might not conform well to RCC predictions, which are based on streams draining more forested landscapes. Regardless of the mechanisms, our investigation indicates that standard functional structure metrics (e.g., functional group ratios) may not be appropriate for assessing these highly modified systems.

Allocthonous organic materials in the form of CPOM from adjacent terrestrial habitats are generally the main energy source for stream communities draining forested landscapes (Fisher and Likens, 1973; Anderson and Sedell, 1979; Cummins et al., 1989; Wallace and Webster, 1996). Once in the stream, this material is leached and broken down into smaller particles through a variety of physical and biological process, including the feeding activities of shredders (Anderson and Sedell, 1979; Cummins et al., 1989; Cuffney et al., 1990). The quantity and quality of CPOM available to shredders in a given stream depends on the type of riparian vegetation bordering the stream and the stream's retention ability (Benfield, 1997; Jones, 1997). One unexpected result of our study was that the stream with the lowest riparian forest cover had the highest amounts of total CPOM in the streambed. However, some of this material was derived from crops adjacent to this stream (e.g., corn litter), and it also appeared that CPOM was more abundant in this stream because much of it was buried in silt and did not decompose; this site also had the highest amounts of silt substrates, and during many sampling events substrates removed from the corer contained buried, black CPOM with a color, texture, and odor typical of an anoxic environment.

We know of no other quantitative estimates of intact crop detritus in streams, but the presence of this material could have important implications for stream function. We commonly found intact pieces of corn (including kernels, cobs, and stalks) in two of the three intensively monitored study streams, and this material could be linked to movement of pesticides that are used in crop production into streams. For example, the rapidly expanding use of *Bacillus thuringiensis* toxins in transgenic crops is of particular interest because δ-endotoxins are present throughout the plant tissues during the entire growing season, and insects that ingest them are killed (National Research Council, 2000). Given that many shredder taxa that feed on CPOM inputs in headwater streams are insects, and that some (e.g., Trichopterans) are closely related to target species of Bt toxins such as the European corn-borer (*Ostrinia nubilalis* Hübner), important headwater stream functions such as the decomposition of CPOM could be altered. Our results clearly demonstrate that, along with sediments from crop fields, crop residues enter streams, and future studies should further examine this as a potential mechanism for transport of pesticides and associated impacts on stream detritivores.

Of the many human impacts on these streams that we observed, sedimentation was the most obvious and likely the most important. Suspended sediments and bedload in streams are the largest pollutants by volume in the United States (USEPA, 1994; Waters, 1995). Excessive sedimentation can degrade stream habitats and compromise biotic integrity, particularly in streams that drain agricultural landscapes like those we studied (Cooper, 1993; Waters, 1995). This was evident in our study, as the site with the lowest riparian forest cover, which in turn had the highest percentage of adjacent crops, had significantly higher percent silt substrates, along with significantly higher percent dominant taxon, significantly lower Shannon diversity, higher Hilsenhoff scores, and higher densities of sediment-tolerant macroinvertebrates such as tubificids and fingernail clams. It is well established that sedimentation degrades aquatic habitats and interferes with reproduction, growth, and survival of aquatic organisms, which ultimately compromises biotic integrity (Cooper, 1993; Waters, 1995). Sedimentation from agricultural activities is a common and chronic problem, especially when combined with altered retention and transport of particulate materials in stream channels. In fact, eroded sediments that enter stream channels can be retained for decades, and thus recovery from this disturbance can be slow (Trimble, 1999).

## Rapid Bioassessment

Our results indicate that there is potential for bioassessment in these highly degraded streams, and that the USEPA's Rapid Bioassessment Protocols are useful for distinguishing health of these low gradient, agricultural streams. The modified Hilsenhoff index was the most useful metric for discriminating among the gradient of conditions that we examined. Our results also indicate that percent insect density, percent oligochaetes, and percent active filterers varied predictably across the gradient of conditions that we examined, and these metrics may prove useful for monitoring and assessment programs in agricultural regions. In contrast, taxa richness appeared variable and unreliable for use in these systems because richness was relatively high in some of the more degraded sites due to increased richness of tolerant taxa. The EPT index was also not reliable in these systems because EPT taxa were absent or rare in the streams examined and the few that were present were mostly tolerant representatives of these usually intolerant groups (such as *Caenis* spp., *Callibaetis* spp., and *Baetis* spp.) This is in sharp contrast to other studies. Whiles et al. (2000) found the EPT index was a useful and efficient metric for assessing biotic integrity in agricultural streams in Nebraska, and Lenat and Barbour (1994) reported that the EPT index was the single most reliable metric employed by state biologists in North Carolina. The EPT index has also been shown to reflect changes in stream ecosystem processes associated with anthropogenic disturbance (Wallace et al., 1996).

Local factors in headwater streams, such as riparian characteristics, water chemistry, and in-stream habitat structure, influence macroinvertebrate community structure, and their influences were evident in this study. For example, Hilsenhoff scores were significantly correlated with both orthophosphate concentrations and percent riparian forest. However, the strongest correlation was with in-stream habitat scores, suggesting that in-stream physical habitat may be the most limiting factor for biotic integrity in these systems. In a review of agricultural impacts on streams, Cooper (1993) noted that physical habitat can be an overriding factor influencing the health of these systems. Thus, although the goals of many biological monitoring and assessment programs are to identify water chemistry problems, the important influence of in-stream physical habitat on biotic integrity must be acknowledged in any study and may make distinguishing the influence of other stressors difficult.

Our results indicate that there is potential for using forested riparian buffers to protect and/or improve stream health in this region, as even small amounts of riparian forest were associated with better in-stream habitat quality and biotic integrity. Our results also add to growing evidence that physical habitat may be the most important factor limiting biotic integrity in agricultural streams, suggesting that management of these highly degraded "drainage ditches" that drain the crop fields of the Midwest should focus on improvements to in-stream habitat quality.

## APPENDIX

Land use within each of the 15 subwatersheds examined in the Sugar Creek watershed. Low cover, medium cover, and high cover are the three intensively monitored sites. Land cover for some watersheds is less than 100% because of small amounts of pasture, bare land, water bodies, and other minor features in them.

| Watershed | Watershed area | Row crop in watershed | Forest in watershed | Urban in watershed | Forest in a 15-m stream buffer |
|---|---|---|---|---|---|
| | ha | % | | | |
| Low cover | 220 | 99 | 1 | 0 | 5.8 |
| Medium cover | 369 | 93 | 4 | 3 | 21.5 |
| High cover | 250 | 97 | 3 | 0 | 31.0 |
| 4 | 509 | 97 | 1 | 2 | 7.5 |
| 5 | 307 | 100 | 0 | 0 | 1.7 |
| 6 | 395 | 86 | 14 | 0 | 1.2 |
| 7 | 625 | 92 | 8 | 0 | 37.0 |
| 8 | 2286 | 87 | 10 | 3 | 31.7 |
| 9 | 297 | 98 | 2 | 0 | 27.8 |
| 10 | 359 | 97 | 3 | 0 | 8.2 |
| 11 | 233 | 97 | 1 | 0 | 1.0 |
| 12 | 436 | 93 | 2 | 0 | 9.5 |
| 13 | 337 | 94 | 0 | 0 | 34.8 |
| 14 | 329 | 96 | 0 | 0 | 38.0 |
| 15 | 412 | 98 | 1 | 1 | 1.0 |

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